GENETIC MARK-RECAPTURE ABUNDANCE ESTIMATE AND DIETARY PREFERENCES IN A RECENTLY RE-ESTABLISHED AMERICAN BLACK BEAR POPULATION

By

EMILY J. ARTZ

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Thesis Approved:

Dr. W. Sue Fairbanks

Thesis Adviser

Dr. David M. Leslie, Jr.

Dr. Barney Luttbeg

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Abstract: American black bears (*Ursus americanus)* are recolonizing parts of eastern Oklahoma that they have not occupied since the late 1800s. The Ozark region black bears are adapting their diets and foraging behaviors to include anthropogenic foods, causing property damage and human-bear conflict. To effectively manage this likely growing population, a baseline abundance estimate and a better understanding of the Ozark American black bear's foraging preferences is necessary.

Live trapping in 2011–2015 marked 49 individuals. In 2014 I established a grid of 46 barbed wire hair snares to collect DNA for a genetic mark-recapture abundance estimate using microsatellite analysis. Over 5750 sampling opportunities in 2014 and 2015, we collected 369 hair samples, 85 of which amplified at ≥ 7 loci. Using a full-likelihood model, we estimated a population of 100.91 individuals ± 23.45 (95% CI). The sex ratio of live captured bears was 2.4 males : 1 female, and the median age of both males and females was 3 years. The young, male-biased demography suggests that the Ozark population contains many recent immigrants dispersing from the larger Arkansas population.

To examine diet preferences, we developed a novel experimental approach to examine black bears' preference for natural versus anthropogenic foods using "giving up density" methods. We designed experimental feeders which were tested on a captive female grizzly bear at the Tulsa Zoo, Tulsa, Oklahoma. Feeders were filled with either natural foods (acorns or blueberries) or anthropogenic foods (corn) and deployed them as pairs suspended from tree branches. The bears manipulate the feeders to obtain a food reward. We standardized the size of the food reward by partitioning food in edible gelatin capsules. Each food capsule contained a standard number of calories regardless of food type. We calculated which food the bears preferred when both were equally available, based on how much of each food type remained after a bear interaction. We deployed pairs of feeders in six locations and monitored them with trail cameras. Over 3 months, bears interacted more often with feeders containing corn, removed more corn, and spent more time manipulating feeders for corn. This suggested that bears were willing to spend more effort to obtain corn than native foods when both are equally available.

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CHAPTER I

INTRODUCTION

Recolonizations by large carnivores are rare events (Hellgren et al. 2005), but there are several examples of large carnivores dispersing into highly altered, human–dominated landscapes (Hellgren 1993; Marucco and McIntire 2010; Scheick and Mcown 2014). Mountain lions (*Puma concolor)* are currently re-expanding their distribution from remnant source populations into parts of the Midwest, using stepping-stones of suitable habitat to establish new viable populations in areas from which they were extirpated in the early 1900s (Larue 2012). Gray wolves (*Canis lupus)* have successfully recolonized parts of Wisconsin and Montana following the recovery of remnant populations after the species was protected under the United States Endangered Species Act (Mladenoff 1999). European gray wolves have also successfully recolonized parts of their historic distribution in the western Alps, using the Italian Apennine Mountains as a dispersal corridor (Marucco and McIntire 2010). Some carnivores have recolonized new areas following successful reintroductions, such as the wolves reintroduced to Yellowstone and Grand Teton national parks (Berger and Gese 2007) and the American black bears (*Ursus americanus)* reintroduced to Arkansas in the 1960s (Smith and Clark 1994).

Rates of human–wildlife conflict will increase as animal populations expand.

For example, the impact of recolonizing wolves on livestock and human safety in parts of Europe and North America has been a subject of major concern in recent decades (Beckman and Berger 2003; Ciucci et al. 1993;). Mountain lions, black bears, and grizzly bears (*Ursus arctos)* face similar conflicts, especially in areas where they have been extirpated for a long time. Negative human attitudes can be a major barrier to recolonizing carnivores (Davenport et al. 2010).

To successfully manage recolonizing species and mitigate human–wildlife conflict, managers must establish baseline data about the size and distribution of a recolonizing population, as well as its potential for growth and continued range expansion (Maehr et al. 2001; Swenson et al. 1998). Studying large carnivores, especially solitary ones, is often difficult due to their large home range and potential for long-distance movement (Mowat and Strobeck 2000). The low density of a newly established population often compounds these difficulties and makes traditional methods such as live trapping for mark-recapture abundance estimation very difficult (Gardner-Santana 2007; Kendall et al. 2008). With recent improvements in PCR processing and DNA analysis, non-invasive genetic mark-recapture methods are becoming more and more widely used for population density analysis (Boulanger et al. 2008; Kendall et al. 2009; Triant et al. 2004).

As recolonizing carnivores expand their range and adapt to human dominated landscapes, they often rapidly alter their behavior to take advantage of the anthropogenic food sources that are now highly available (Beckman and Berger 2003; Mulhy and Musiani 2009). Omnivorous species often incorporate garbage into their diet (Beckman

and Berger 2003), and carnivores preying on livestock is a source of major human–wildlife conflict (Greenleaf et al. 2009; Mulhy and Musiani 2009).

Recolonizing carnivores may use anthropogenic food sources to compensate for natural foods becoming less available (Greenleaf et al. 2009). They may also seek out anthropogenic foods in preference to their natural diet because they provide better nutrition or energy, or because they are convenient and abundant and more easily accessible than natural diet sources (Ditmer et al. 2016). Without a direct comparison between equally available diet types, we do not know whether a species expressing a preference for natural or anthropogenic foods.

Giving-up density (GUD) is a tool used to determine diet preference (Brown 1988). It is calculated by providing an animal with food items mixed in substrate, so that they have to expend time and effort to extract the food. This creates an effect of diminishing returns: after a certain point, the benefits of the food no longer outweigh the cost of obtaining it, and the animal will give up and leave the patch: the density of food remaining when the animal leaves is a GUD (Charnov 1976). GUD has been used to study patch use and perceived predation risk by prey species in a variety of small granivorous rodents (*Perognathus amplus*, *Diopodomys merriami*, *Ammospermophilus harrisii*, and *Spermophilus tereticaudus*) (Brown 1988) and grazing ungulates such as the Nubian ibex (*Capra nabania*) (Iribarren and Kotler 2012). GUD can also be adapted to study food preference by offering different food types that are equally available: the lower the GUD, the greater the animal's preference for that food. This method has been used very effectively to investigate diet preference in Samago monkeys (*Cercopithecus mitis erythrarchus*), omnivorous mesopredators with highly variable diets (Emerson and

Brown 2012), but the method has never been adapted to examine diet preference in carnivores.

The recently established bear populations in the Ozark region of eastern Oklahoma exemplify challenges that face both recolonizing carnivores and the biologists attempting to manage the expanding population. After being extirpated from the majority of their historic distribution in the southeastern United States (Dobey et al. 2005), black bears are gradually re-expanding into areas they have not occupied since the late 1800s (Tyler and Anderson 2009). Following an extremely successful reintroduction of black bears to Arkansas in the 1960s (Smith and Clark 1994), black bears began dispersing westward into Oklahoma and established a viable breeding population in mid-1990s (Bales et al. 2005). A new population growing in a significantly changed part of its historic distribution presents a variety of new challenges in management and conservation of the species.

The black bear population in the southeastern Ouachita region of Oklahoma is well established (Bales et al. 2005; Pfander 2016). However, the northern Ozark region of Oklahoma has only been recolonized in the last 10–15 years by Arkansas bears dispersing across the state line. An accurate population estimate is not currently available for the region (Bales et al. 2005). Because the recolonization of human-dominated areas will increase the rate of human-bear conflict (Baruch-Mordo et al. 2008; Beckmann and Berger 2003), an understanding of black bears' use of anthropogenic resources like human foods and urban denning sites is needed.

In the United States, the frequency of human-bear interactions has been increasing since the 1960s, as human population expands and bears are forced to adapt to highly altered environments (Merkle et al. 2011). Black bears are omnivores with highly adaptable diets (Fortin et al. 2012), and they will often expand their diets to include a large percentage of anthropogenic foods (Hopkins et al. 2012). In other areas with thriving bear populations, this has been a source of significant human-bear conflict (Greenleaf et al. 2009) and is therefore a major source of concern for managers. In Oklahoma, many land owners have reported damage caused by bears breaking into deer feeders for corn. It is not clear, however, whether black bears expand their diets to include anthropogenic foods because their natural food sources are less available, or because they actively prefer anthropogenic foods to their natural diet. To manage bear populations in these altered environments and mitigate potential human–bear conflict, baseline knowledge about the re-establishing population and its resource use in a new environment is needed.

CHAPTER II

POPULATION DEMOGRAPHICS AND ABUNDANCE ESTIMATES USING MICROSATTELLITE ANALYSIS OF DNA COLLECTED FROM HAIR SNARES

Estimating abundance of any population is critical to their conservation and management, especially in low–density or isolated populations that are vulnerable to inbreeding or local extinctions (Triant et al. 2004). Abundance estimates are often difficult for populations of large carnivores that tend to be widely distributed, naturally evasive, and often inhabit dense and difficult to access habitats (Mowat and Strobeck 2000; Pelton 1991; Poole et al. 2001; Triant et al. 2004). The live capture and handling of large carnivores are also risky, both to researchers and animals (Mowat and Strobeck 2000; Poole et al. 2001).

Significant improvements in PCR and DNA analysis technology allow development of non-invasive sampling techniques that work well for sparse populations. Modern PCR allows amplification of tiny amounts of DNA (Durnin et al. 2007; Morin et al. 2001), and several studies have used PCR and microsatellite analysis to genotype individuals using DNA collected from hair follicles. The individual genotypes serve as genetic marks and recaptures, making it possible to accurately estimate the

abundance of large carnivore without the need for risky, expensive, and time consuming live captures.

Hair snare sites are easier to set up and can be checked less frequently than live traps, allowing surveys in large areas simultaneously. The simpler logistics and larger potential sample size of hair snare surveys also allow for study designs that better match the statistical assumptions of mark-recapture models (Poole et al. 2001). Hair samples collected from baited hair snares have been used to successfully estimate the abundance of populations of elusive carnivores such as marten (*Martes americana;* Foran et al. 1997) mountain lions (*Puma concolor;* Russell et al. 2012), and many populations of grizzly bears (*Ursus arctos;* Boulanger et al. 2002; Boulanger et al. 2004; Kendall et al. 2008; Kendall et al. 2009; Mowat and Strobeck 2000; Poole et al. 2001) and American black bears (*Ursus americanus;* Belant et al. 2011; Boersen et al. 2003; Garshelis and Noyce 2006; Settlage et al. 2009; Tredick et al 2007; Triant et al. 2004).

American black bears are native to Oklahoma, but they had been entirely extirpated from the state by 1915 (McCarley 1961). Their extirpation was the result of two main factors: unregulated harvest and habitat destruction, most notably clearcutting of the Ozark Plateau in the early 1900s (Smith and Petit 1988).

In 1958, Arkansas Game and Fish began a translocation program to re-establish American black bears in the Interior Highlands of Arkansas, with a goal to reopen a hunting season in the region (Smith and Clark 1994). Historically, translocations of carnivores have had limited success, but the Arkansas translocation remains the most successful such attempt in the world. The population increased from 254 animals to an

estimated 2500 over the next 20 years, and by the late 1980s, American black bears began expanding into the Ouachita Mountain region of southeastern Oklahoma (Bales et al. 2005; Skeen 1997). In the late 1990s and early 2000s, bears began to recolonize the Ozark Plateau region of eastern Oklahoma as well.

While the Ouachita National Forest has a large area of public wildlands containing very productive bear habitat, the Ozark region of Oklahoma is primarily small patches of private land used for agriculture (Diamond and Elliot 2015). The black bears currently recolonizing this area are integrating into a highly fragmented, human-dominated environment, and often seek out anthropogenic sources of food to supplement their diet, causing property damage and increasing rates of human-bear conflict (Artz 2016, unpublished data). This has influenced increased demand for an American black bear hunting season in the Ozark region. A season already occurs in four counties in the Ouachita Mountain region of southeastern Oklahoma.

Research on the population of black bears inhabiting the Ozark region began in 2011 with live trapping, marking, and satellite collaring. While live trapping provided valuable information on the distribution and demographics of the recolonizing population, the number of live captures between 2011 and 2015 was not sufficient for an accurate abundance estimation. The ongoing project was expanded to include hair snare sampling for a genetic abundance estimate in 2014. Our objective in this study was to obtain an accurate abundance estimate of the Ozark region black bear population. This estimate will serve as a baseline to inform current and future management decisions.

Methods

Study area

Our study area covered parts of 3 counties in eastern Oklahoma (Cherokee, Sequoyah, and Adair), which encompass the part of the Ozark Plateau that extends into the state (Fig. 2.1). The core area of black bear habitat in the Oklahoma Ozarks extends from the eastern edge of Lake Tenkiller to the Arkansas/Oklahoma state line. The majority of the area is privately owned. Public lands are either designated for recreational use or managed for deer and elk hunting by the Oklahoma Department of Wildlife Conservation (ODWC) and the United States Army Corps of Engineers.

The study area includes the ODWC's Cookson Hills Wildlife Management Area (WMA), a 5,960-ha management area in southeastern Cherokee and southwestern Adair counties. The WMA is characterized by rolling, rocky hills and dense stands of oak-hickory timber, with some short-leaf pine at higher elevations, and is home to the highest density of bears in the study area (Sara Bales Lyda, pers. comm.).

Hair collection

Hair snares were set in a grid of 48 cells across the core area. Each cell was 4.8 km^2, the approximate size of a female black bear's home range in this area (Bales et al. 2005). We placed barbed wire hair snares as close as feasible to the center of each cell. When deciding where to place snares, we looked for bear trails, scat, and nearby water

sources or travel corridors. We also used information from live trapping studies, trail cameras, and local reports of bear activity.

Snares were constructed with two strands of four-barbed wire, stretched around trees at 25 and 60 cm off the ground and tightened with standard fencing tools (Brown 2008; Woods et al. 1999). We baited each snare with a can of sardines suspended in the center of the snare with cotton string, and a tampon soaked in raspberry or maple extract to act as a scent lure (Settlage et al. 2008). We replaced baits every time the snares were checked and tightened wires as needed. We used 15 motion activated Stealth Cam model STC-G42NG trail cameras (Stealth Cam LLC, Grand Prairie TX) to monitor snares. Because we had a limited number of cameras available, they were moved throughout the sampling periods, to monitor high-activity sites or troubleshoot problem sites.

We checked each snare every 7–10 days to minimize sample loss and degradation due to exposure (Foran et al. 1997; Taberlet et al. 1997), although road conditions and landowner requests sometimes lengthened return times. We considered every day that an individual snare was baited to be a sampling opportunity, analogous to trap nights in a live-trapping study. We collected any hair using forceps and gloves and recorded snare number, barb number, date, and species. We flame sterilized forceps and wire barbs with a butane lighter to prevent cross contamination (Brown 2008). We also collected hair opportunistically from natural rub trees and telephone poles in the sampling grid (Boulanger et al. 2008). The samples were dried in desiccant for 24 hours to prevent molding, and then frozen at -20 °C (Mowat and Strobeck 2000).

We checked hair snares weekly between 2 June and 11 August 2014, and monthly thereafter through 12 December when satellite collared black bears in the region had denned for winter hibernation. In 2015, we repaired and re-baited hair snares and checked them weekly between 14 May and 12 August, at which point the snares were disassembled.

Live trapping

Black bears were also live trapped in summer 2014 and 2015 using full enclosure barrel or culvert traps baited with pastries, sardines, and feed corn. The live trapping team maintained trap lines of 7–9 traps for 3–4 weeks at a time, checking all traps daily. We sedated captured bears with a 2:1 mixture of Telazol and Xylazine (Clark and Smith 1994), administered intramuscularly with a pole syringe (Clark 1991). The captured bears were marked with plastic ear tags and a lip tattoo with corresponding unique identifying numbers. The tissue removed to attach ear tags was reserved for future analysis. We also collected a first upper premolar for aging (Costello et al. 2004) and a hair sample for DNA analysis.

DNA extraction from hair samples

We harvested follicles from hair shafts using a razor blade and dissecting microscope. All equipment was cleaned with 90 percent ethanol between uses to prevent cross contamination. Between 5 and 27 follicles, preferably from guard hairs (Gagneux et al. 1997; Goossens et al. 2002; Taberlet et al. 1997; Triant et al. 2004) were transferred to a sterile microcentrifuge tube and re-frozen at -20 °C. We cut the hair shaft as close to the follicle as possible to avoid the PCR-inhibiting melanin in the shaft itself.

DNA was extracted from hair samples using the Chelex method (Walsh et al. 1991). Follicles were placed in 200 µL of a 5 % Chelex (BioRad, Hercules, CA) solution and incubated overnight at 56 °C. The incubated samples were vortexed for 10 seconds, heated at 100 °C in a heat block, vortexed for a further 10 seconds, and microcentrifuged for 3 minutes at 13,000 rpm.

PCR and identification of individual genotypes

Each hair sample was genotyped for two dinucleotide microsatellite loci developed from black bears (G10L and G1D; Paetkau 2004) and 7 tetranucleotide loci, also developed from black bears (EU031659, EU031665, EU031691, EU031692, EI031726, EU031698, EU031708; Sanderlin et al. 2009).

We took several steps to remove melanin and other potential contaminants from the DNA extract before PCR. The DNA extract was processed through the OneStep™ PCR Inhibitor Removal Kit (Zymo Research, Irvine CA). We pipetted the DNA extract into the filter and centrifuged it at 8,000 rpm for 1 minute to remove contaminants. We used a Nanodrop spectrophotometer to estimate concentration of nucleic acid in each sample extract before PCR. Due to the low number of follicles on many samples, each sample was run through the PCR process twice to increase the amount of genetic material available for analysis.

Reactions were conducted in an MJ Research, Inc. PTC 100® Programmable Thermo Cycler (Bio-Rad Laboratories Inc., Waltham, MA). Each 15µL reaction contained 1.8µL True Allele PCR Premix (Applied Biosystems, Foster City CA), 0.1 mM primer pair, a variable amount of Chelex extraction based on the Nanodrop concentration,

and a distilled water buffer to bring the total volume to 15µL. For the dinucleotide primers (G1D and G10L), the thermal profile was: 95 °C for 10 minutes, 45 cycles of 95 °C for 30 seconds, 57.5 °C for 30 seconds, 72 °C for 40 seconds, 72 °C for 10 minutes, and finally 4 °C for 2 minutes (Gardner-Santana 2007). For the tetranucleotide primers, the thermal profile was 21 cycles at 96 °C for 20 seconds and 60 °C for 30 seconds, 72 °C for one minute, 14 cycles of 96 °C for 20 seconds, 50 °C for 30 seconds, and 72 °C for 1 minute, followed by a final extension period at 72 °C for 10 minutes (Sanderlin et al. 2009). We then incubated the primer mix for 2.5 hours. Following incubation, we added 1.2µL of the PCR product to 9.0µL of HiDi 2x PCR Master Mix polymerase and 0.5µL ROX reference dye. We scored individual genotypes by hand using GENOTYPER software.

We used GENECAP 1.4 software (Wilberg and Dreher 2004) to detect matching genotypes, allowing for mismatch at one locus. The majority of mismatches were due to loci that failed to amplify. We calculated probability of identity (PI; Paetkau 2004) and probability of siblinghood (PI*sibs*; Waits et al. 2001) using GenAlEx (Peakall and Smouse 2012) and CERVUS software (Kalinowski et al. 2010). PI is a statistical measure that represents the probability of two randomly selected samples having the same genotype due to coincidence, rather than originating from the same individual. PI decreases as more loci are incorporated into the analysis, and can be used to determine the minimum number of amplified loci needed to accurately identify individual genotypes (Paetkau 2004; Waits et al. 2001). We accepted values of PI < 0.0001 (Waits et al. 2001): the probability of two samples sharing a matching genotype due to coincidence rather than being from the same animal is less than 0.0001 for a given

number of amplified loci. PI*sibs* is a more conservative estimate of the same probability, incorporating the probability of two samples having the same genotype due to siblinghood as well as random coincidence. We accepted values of P*sibs* < 0.05 as a unique genotype (Mowat and Strobeck 2000; Woods et al. 1999), based on a mean litter size of 2 in the eastern Oklahoma region (Bales et al. 2005). We estimated abundance of bears using a full likelihood mark-recapture model (Otis et al. 1978) in program MARK (White 2008). We assumed no or negligible emigration or immigration during the study due to our short sampling sessions and small study area (Brown 2008). We calculated full likelihood estimates based on both genetic mark-recapture and the encounter history of bears live captured between 2011 and 2014 to allow for comparison between the two methods.

Results

Hair capture, extraction, and amplification

In 2014 and 2015, we collected a 369 bear hair samples from 46 hair snares (156 in 2014, 213 in 2015). These samples were collected across 5,750 sampling opportunities (1,994 opportunities at 36 snares in 2014, 3,806 opportunities at 46 snares in 2015). All bear hair samples were collected from 23 of the 46 snares, and more than 80% of samples (295) were collected from a cluster of 6 snares located in and around the Cookson Hills WMA. Of the 369 bear hair samples collected, 242 had ≥ 3 follicles. We were successful in extracting DNA from 152 (41%) samples, including samples from 12 live captured bears. Eighty-five of these samples amplified at 7 or more loci during genetic analysis and were included in the population estimation models. In some cases, multiple samples

collected during a single session were from the same individual. In these cases, we considered the sample a single encounter.

Genetic analysis identified 63 unique genotypes, representing 63 individual bears. The error rate when scoring genotypes was 8%, within acceptable limits (Gardner-Santana 2007). Combining live captured individuals (12 bears) and individuals genetically identified from hair samples, we "marked" 75 individual bears.

We recorded 24 samples that matched a previously genotyped bear at ≥ 7 loci and were considered recaptures. The majority of these pairs consisted of two encounters: one mark and one recapture (Fig 2.2). One individual, a young male also first identified as a nuisance live capture, was encountered 8 times at 7 different snare sites across both years. Only 2 of 12 black bears live-captured and genotyped in 2014 and 2015 were identified in hair snare captures.

A full-likelihood mark-recapture model run in program MARK estimated a population of 100.91 animals ± 23.45 (95% CI). For combinations of 5 or more amplified loci, PI = 0.0000012 and PI*sibs* = 0.0067, an acceptably low probability of identity (sufficiently low PI < 0.0001; Waits et al. 2001).

Live capture and population structure

Black bears have been live-trapped in this region since 2011. Fifty-one individuals have been tagged: as of 2015; 49 are still alive. In 2014 and 2015, when the trapping seasons corresponding with the hair snare study, we had 989 trap nights (563 in 2014, 426 in 2015). In 2014, we captured 15 black bears, 6 of whom were previously unmarked. In 2015, we captured 34 individuals, 10 of whom were previously unmarked.

Five of those individual were born in the study area and tagged with PIT tags during routine den visits. All the previously unmarked captures in 2015 were young males.

A similar full-likelihood model using the encounter histories of 49 bears live captured between 2011 and 2015 estimated a population of 69.71 animals ± 13.24 (95% CI).

Sex ratio of the population was biased male, with 36 tagged males to 15 females, a ratio of 2.4:1. The oldest male in the captured sample was 11 years old and the oldest female was 8 years old, but the population is biased toward subadults and juveniles (Fig. 2.3), with a median age of 3 years for both males and females. The population has a relatively high fecundity of 0.8, with 10 reproductive mothers producing 8 female cubs from 2011–2015.

Spatial patterns of bear presence

We used rates of activity at hair snares, defined as the percentage of sessions that snare was checked that hair was present, to provide a visual representation of areas with high rates of activity, and presumably a high density of bears (Fig. 2.4). The hair snares with the highest rates of activity were tightly clustered at Cookson WMA in the northwestern corner of the core study area, where more than 80% of the hair samples used in this study were collected. There is a secondary activity hotspot in the south-central area of the grid, in and around several contiguous sections of private land. 24 snares in the western and northeastern edge of the study area had no activity. Plotting rates of hair snare activity across the sampling grid indicates a potential corridor running

from the Arkansas/Oklahoma border to the Cookson Hills WMA, the largest area of public, managed land in the study area (Fig 2.4).

Discussion

The results of our genetic mark-recapture study suggest that the population of black bears in the core area of the Oklahoma Ozarks contains 100.91 ± 23.45 individuals, and the majority of the population consists of young males < 5 years old. Nevertheless, analysis problems such as failure of loci to amplify or allelic dropout (Paetkau 2004) can cause duplicate samples to be counted as more than one individual, leading to an overestimation of abundance (Creel et al. 2003; Mills et al. 2000; McKelvey and Schwartz 2004a; Waits et al. 2001; Waits and Paetkau 2005). These problems are especially prevalent in microsatellite analysis of hair snare samples, where the tiny amounts of DNA being tested can lead to high rates of amplification failure and allelic dropout (McKelvey and Schwartz 2004b). The potential for overestimation of abundance must be taken into account when making management decisions based on genetic abundance estimates, especially in a small population in a small geographic area.

The small size of this population and its clustered distribution also presented challenges in obtaining sufficient data for analysis, despite the advantages of hair snares and other non-invasive sampling methods in low density populations. Although we had genotypes for 13 bears live captured in 2014 and 2015, only two of them were detected in samples collected from hair snares. While non-invasive sampling methods can cover a large area with relatively low cost and effort, it is difficult to control the quality of samples collected. In previous hair snare studies of black bears, 40–60% of hair samples

collected did not contain enough DNA to amplify (Brown 2008; Gardner-Santana 2007). Large-scale studies can collect thousands of samples to compensate for losses to insufficient DNA or contamination. In small-scale studies, contamination or failure to amplify becomes a significant problem, because the loss of 40–60% of samples can result in a sample size too small for accurate statistical analysis. With only 150 amplifiable samples in this study, the use of tetrad primers was very valuable, because they allowed for much more accurate scoring (Sanderlin et al. 2009): 56% (85 of 150) of amplified samples could be scored at ≥ 7 loci with an 8% error rate, a higher success rate than previous studies achieved with diad primers in eastern Oklahoma black bears (Brown 2008; Gardner-Santana 2007).

The small sample size of small-scale studies can also lead to a statistical inaccuracy, as the sample is less likely to be representative of the population. We believe this is the reason that we recaptured only two previously live-captured bears in hair snare samples. This is likely an erroneously low recapture rate, not representative of the proportion of marked bears present in the population. Trail camera photos indicate that other previously live captured bears visited hair snares, but any hair samples they left did not contain enough DNA or failed to amplify during genetic analysis.

While obtaining enough samples for a significant analysis can be a challenge in small-scale hair snare studies (Boulanger et al. 2002; Boulanger et al. 2008), they still offer other valuable information and potential insights. Hair snares and other non-invasive sampling methods are an efficient and low-cost way to cover a large area, especially if the target species is rare or patchily distributed (McKelvey and Schwartz 2004b). Because of the expense and time investment required to run live traps, they need

to maintain a reasonably high capture rate to be worth the effort expended. Hair snares can be left baited for 7–10 days between visits and require less time and money to construct and maintain, so inactive snares are not a problem like inactive traps. Hair snares also provide more complete and useful information on animal distribution, even in low activity areas. Analysis of the rates of activity on hair snares in the Oklahoma Ozarks shows a very clumped distribution, with the majority of the population located on and around the Cookson Hills WMA.

Activity levels and spatial presence/absence data provided by the hair snares proved to be useful independent of the genetic data from hair snares. The large grid of snares sampled across 2 years provide2 a more even picture of the population's spatial distribution than the frequently moved live traps. Hair snare activity can be used to locate sites for future live capture studies. Even hair snares with zero activity across both years provided information into the areas and habitats that the Ozark bears avoid.

This clumped distribution is consistent with expectations for population distribution and habitat use in a human-dominated landscape, where the majority of productive bear habitat is fragmented in a matrix of urban and rural homes or pasture and farmland (Diamond and Elliot 2015). The Ozark bear population is gradually integrating into a human-dominated environment, most likely driven by immigration from the larger population in the Arkansas Ozarks. The demography of the population is heavily skewed toward young males, which are likely dispersing from their natal range in Arkansas into the less densely occupied habitat in Oklahoma. In 2015, all unmarked bears captured in the live traps were males aged 2–3. Five of these individual were born in the study area, and the remaining five were likely immigrants from the Arkansas population. This

suggests that dispersal from Arkansas has a significant impact on the demography of this population. However, it is difficult to estimate a carrying capacity for the region due to a lack of detailed spatial and habitat analysis. A detailed habitat analysis will be essential for future management decisions as the population continues to grow.

Management Recommendations

While genetic estimates set the population size at 100.91 $\pm$ 23.45 bears, the young and male-biased demography of this population suggested that it was not yet well-established in the area. The low number of adult females in the population indicates that population growth due to reproduction is likely to be slow, and any mortality of reproductive-aged females would further depress population growth. Our results indicate the importance of basing management decisions, such as institution of a hunting season, on population demography in addition to population size. At the time of this study, the population was centered in the relatively small area of the Cookson Hills WMA. Further study of black bear habitat use in this region and avoidance of areas of human concentration or activity will be needed to assess the availability of resources for further population growth and expansion. Knowledge of human attitudes will also be needed, because the population is likely to reach a social carrying capacity (Bowman et al. 2001; Davenport et al. 2010) before it reaches an ecological carrying capacity.

Due to the young male biased demography of the population, opening a hunting season would pose a considerable potential risk to the population's ability to persist. The harvest of one of the few reproductive aged females would have a strong negative impact on the population's overall fecundity and rate of growth. If a hunting season is opened, it

would ideally be a limited draw, to avoid overshooting an opening day quota and harvesting too many animals from this small and dispersed population. We also recommend the season be opened late in the year, when reproductive females have denned for the winter and are much less vulnerable to hunter harvest.

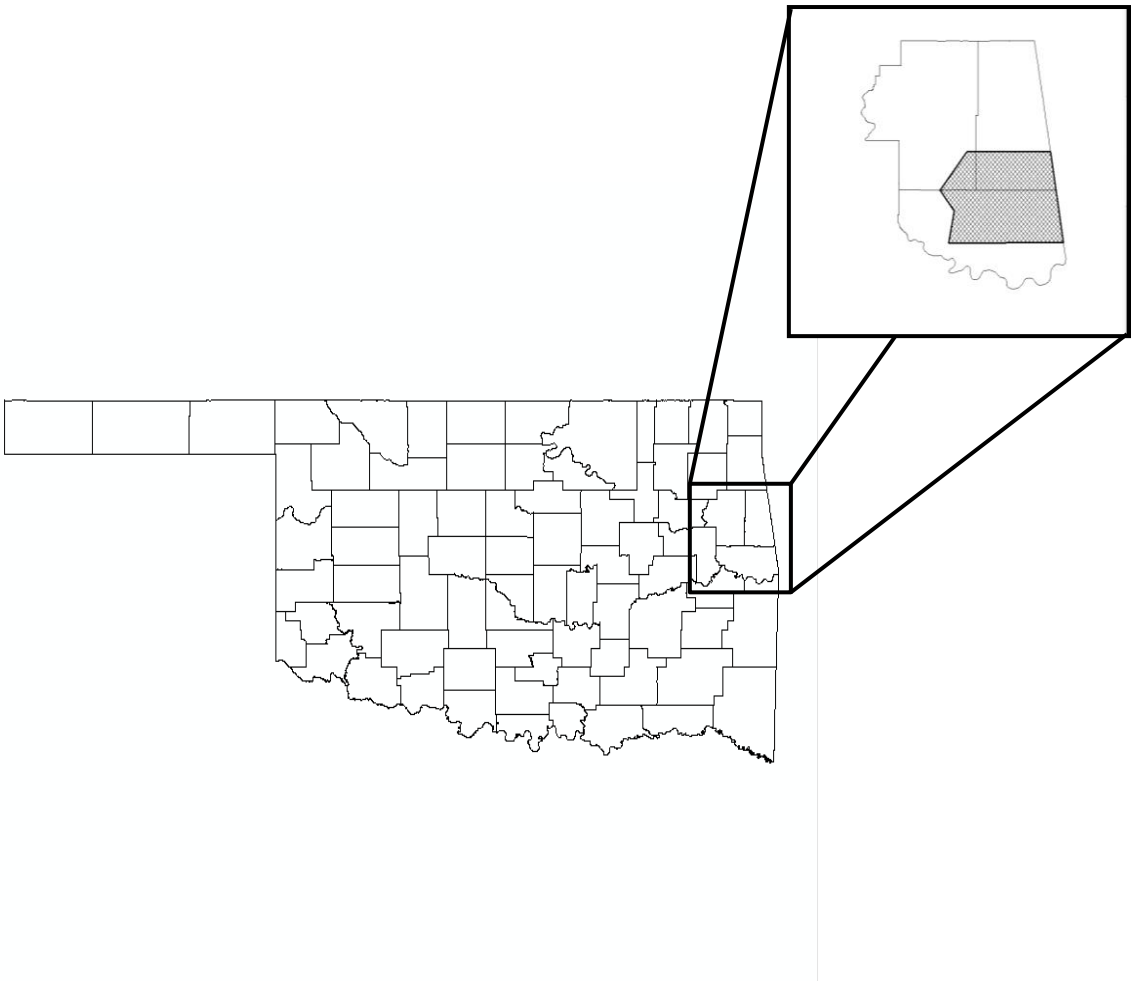

Figure 2.1 American black bear (*Ursus americanus*) study area in east-central Oklahoma, USA, 2011–2015. The core area, shaded in gray, covers parts of Cherokee, Adair, and Sequoyah counties. Live captures were conducted on multiple lines within the core area from 2011–2015. Hair snare sampling took place on a grid within the core area in 2014–2015.

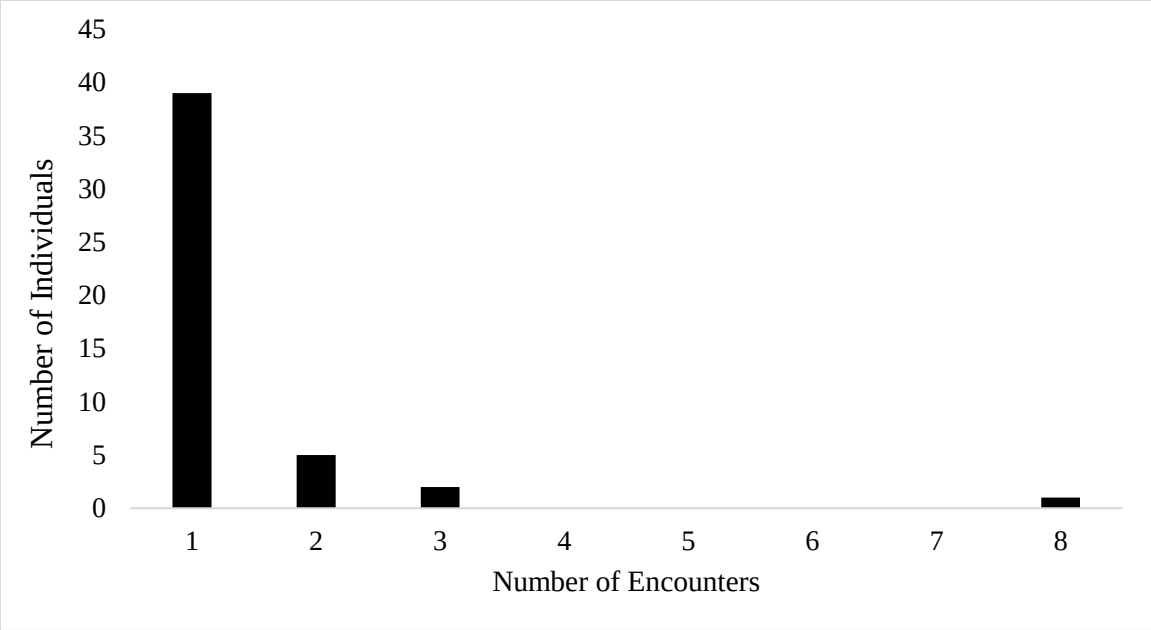

Figure 2.2 Number of encounters for individual black bears (*Ursus americanu)* genotyped at ≥ 7 loci. The x-axis represents how many times an individual was encountered on the hair sampling grid; the y-axis represents the number of individuals encountered a given number of times.

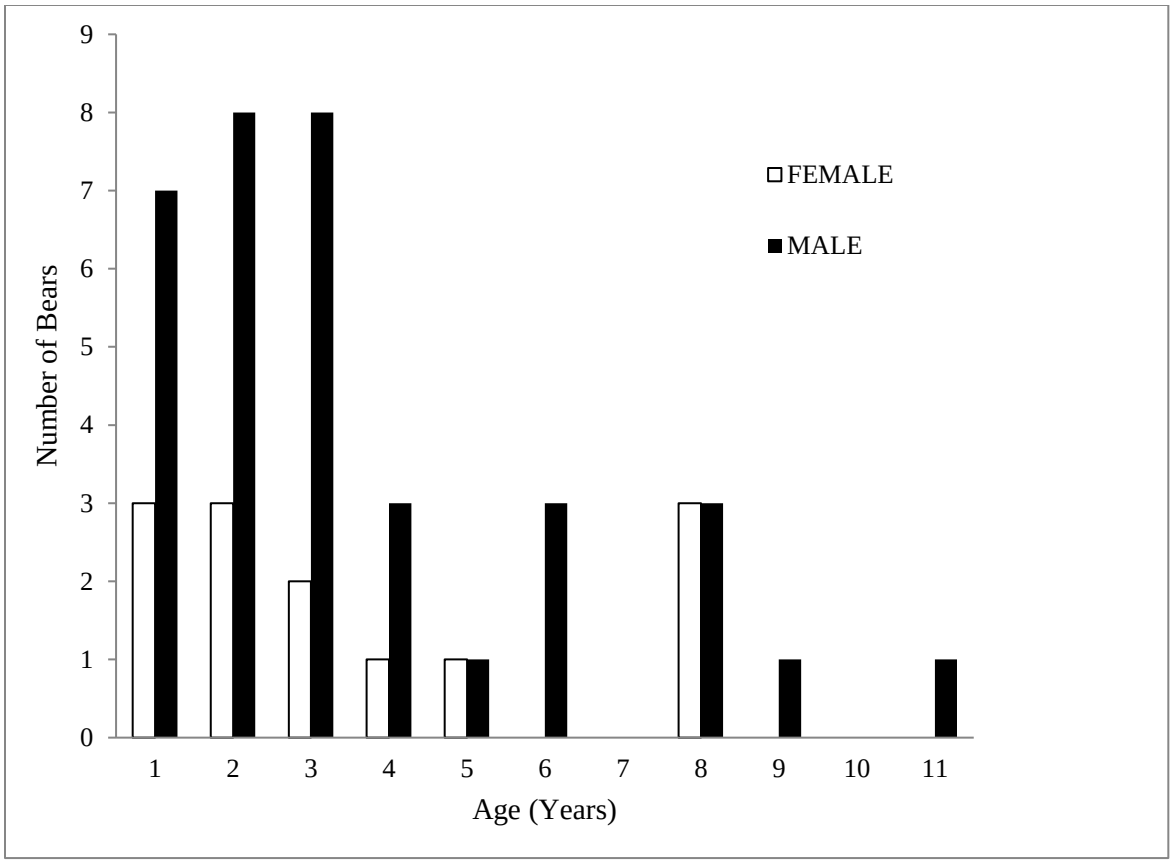

Figure 2.3 Population age structure as determined by cementum annuli analysis of 13 female and 35 male American black bears (*Ursus americanus*) live captured in the Oklahoma Ozark region 2011–2015. (N = 48).

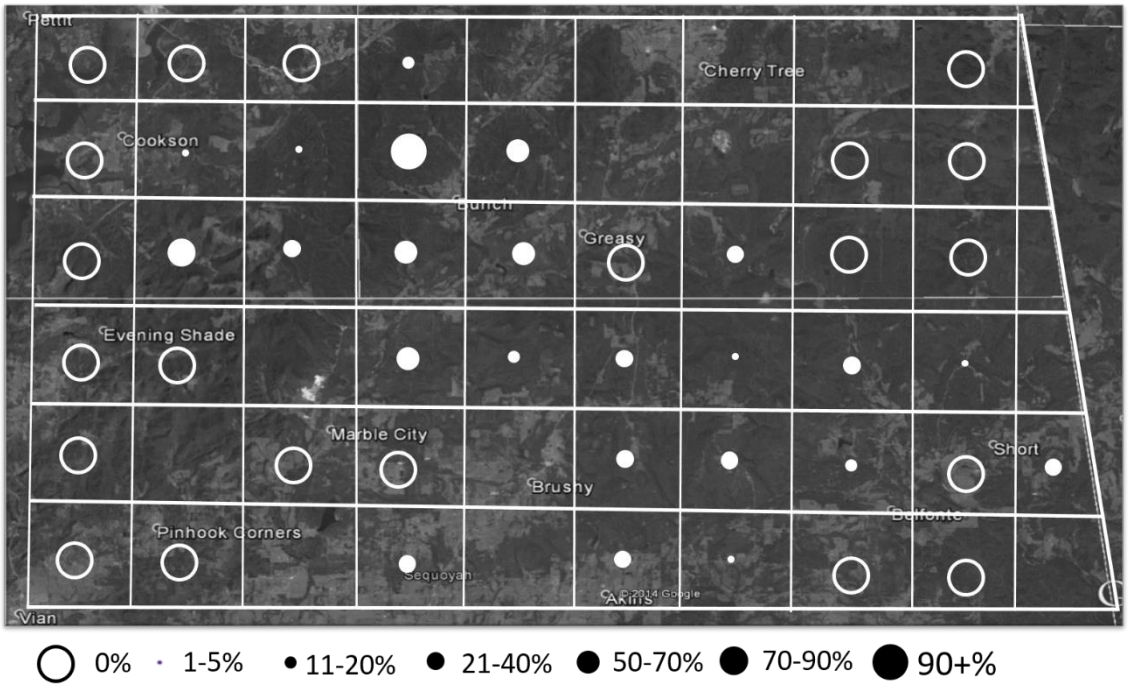

Figure 2.4 Rates of activity at hair snares across the 2014–2015 sampling seasons. Percentages represent the proportion of sessions in which that snare was checked and bear hair was present. Open circles represent a grid cell in which a snare was present, but no samples were collected.

CHAPTER III

ADAPTING GIVING UP DENSITY TO STUDY DIET PREFERENCE IN A RECOLONIZING POPULATION OF BLACK BEARS

As of 2008, one-half of the world's human populations lived in urban centers. This trend is projected to continue, with fringe urban and suburban areas experiencing some of the highest rates of growth in the world (Baruch-Mordo et al. 2014). In many parts of the United States, massive alterations to the landscape have forced wildlife species to adapt their behaviors to urbanized and human-dominated landscapes (Western 2001). While human expansion and the resulting loss and degradation of habitat are usually harmful to wildlife (Gehrt et al. 2010), some species, including some large mammals (Beckman et al. 2003; Berger et al. 2007) are able to successfully adapt their behaviors to survive in a human-dominated landscape. Urban and human-dominated areas offer a high concentration of reliable resources that many species find desirable, despite the increased risk of human proximity (Baruch-Mordo et al. 2014).

Generalist mesocarnivores, such as raccoons (*Procyon lotor*), coyotes (*Canis latrans*), and red foxes (*Vulpes vulpes*), and herbivores, such as white tailed deer (*Odocoilus virginianus*), have adapted as urban areas expand into their native distributions, using the abundance of anthropogenic food sources available in areas with

dense human populations (Gehrt 2010). Other species use anthropogenic resources to expand their distribution. For example, sheds, porches, and building foundations provide winter cover for Virginia opossums (*Didephis virginiana),* allowing them to expand their historic distribution northward into parts of New England that they did not previously occupy (DeStefano and DeGraaf 2003).

Reintroductions and changes to wildlife and environmental regulations have facilitated natural recolonizations of wildlife species back into their historical distribution. Mountain lions (*Puma concolor*) are currently re-establishing in parts of their historical distribution from remnant source populations into parts of the Midwest, using stepping stones of suitable habitat to establish new viable populations in areas from which they were extirpated in the early 1900s (Larue et al. 2012). Gray wolves have successfully recolonized parts of Wisconsin and Montana following reintroduction in Wyoming and recovery of remnant populations after the species was protected under the Endangered Species Act (Mladenoff et al. 1999). American black bears (*Ursus americanus*) were reintroduced to western Arkansas in the 1950s and 1960s, and are now recolonizing Missouri and Oklahoma (Bales et al. 2005, Gardner-Santana 2007).

Recolonizing carnivores with omnivorous diets are capable of rapidly adapting their dietary preferences and foraging habits to include anthropogenic foods (Beckmann and Berger 2003). Crops and foods used to bait other species of wildlife (especially corn fed to white-tailed deer; hereafter referred to as deer corn) offer large omnivores a reliable source of food that is high in calories and fat (Hellgren et al. 1993; Gray 2003) and not affected by year-to-year variations in availability and quality like natural hard and soft mast crops (Howe et al. 2012; Ditmer et al. 2016). Other, more carnivorous

species prey on livestock as an additional source of food. Human activities also have indirect effects: supplemental deer feeding concentrates animals and has been shown to affect the hunting behaviors of cougars (Beckmann and Berger 2003). Similarly, in prey-depleted urban areas of northern Ethiopia, spotted hyenas (*Crocuta crocuta)* concentrate around garbage dumps, scavenging on organic human garbage and nearby livestock (Yirga et al. 2015).

Damage and depredation caused by recolonizing species are a major source of human-animal conflict, especially when recolonizing wildlife damages livestock, crops, or property (Beckmann and Berger 2003; Ciucci et al. 2009). Negative human attitudes can be a significant barrier to recolonizing species, especially large carnivores (Davenport et al. 2010). Surveys indicate that bear damage has a strong influence on the public perception of the size of a bear population and increases demands for mitigation or removal (Garshelis et al. 1999).

Recolonizing species may use anthropogenic food sources to compensate for natural foods being less available (Greenleaf et al. 2009). In productive habitat, use of anthropogenic resources by black bears is a reversible trend that coincides with poor production years; however, in areas of degraded or marginal habitat, the trends of urbanization appear irreversible in black bears (Beckmann et al. 2004). Wildlife may also seek out anthropogenic foods in preference to their natural diet because they provide better nutrition or energy, or because they are convenient and abundant, and more easily accessible than natural diet sources (Baker et al. 2008).

While American black bears are classified as carnivores, their diet is omnivorous and highly adaptable (Beckman et al. 2003). Multiple American black bear populations

have successfully recolonized urban and human-dominated landscapes (Smith and Clark 1994), incorporating large volumes of anthropogenic foods into their diet. Black bears are currently recolonizing parts of their historic distribution in the eastern Oklahoma Ozark mountain region. The region is largely composed of private lands managed for deer hunting or cattle and horse ranching. Abundant blackberries (*Rubus spp.*) are the primary source of natural forage during summer months (J. Connor, unpublished data). There are notable periods of low caloric density in this region: early summer, when soft mast is not ripe and black bears forage primarily on acorns remaining from the previous winter, and late summer and early autumn, when berries are no longer abundant but hard mast has not begun to fall (Ditmer et al. 2016). Feeders containing deer corn are common throughout the area, and many landowners keep them filled year-round to attract deer. These feeders are often exploited by black bears as a reliable source of food throughout the year. Damage to deer feeders is a significant source of human-bear conflict in the region (C. Allen and C. Endicott, pers. commun.) Nevertheless, it is not clear if black bears are using deer corn because they prefer it to their natural foods, or if the concentration of easily obtainable calories at a feeder provides a foraging advantage. Black bears may also use corn from deer feeders as a necessity, because they cannot obtain enough calories from the natural forage available. Because the availability and effort to acquire their natural forage and deer corn vary widely, a different approach is needed to establish whether or not bears exhibit a preference for natural or anthropogenic food sources.

We introduce a novel system that adapts giving up density (GUD) methodology (Brown 1988) for use with black bears, to examine whether black bears in the Oklahoma

Ozarks exhibit a preference for natural or anthropogenic foods when both are equally available. We deployed paired feeders, one containing a known amount of corn and one containing a known amount of a natural diet item, in areas of high bear activity in the Ozark region of Oklahoma. Feeders used to calculate GUD dispense a food reward at a diminishing rate of return, until the animal manipulating the feeder decides the rate of return is no longer worth the time and effort of manipulating the feeder and abandons it (Brown 1988; Garb et al. 2000). We predicted that, given equal access to, and equal effort to obtain, natural and anthropogenic foods, bears would preferentially select their natural diet items over anthropogenic corn.

Methods

Feeder design

To test feeding preferences using GUD methods, we designed a feeder bears must manipulate to obtain a food reward. The feeders mimic the diminishing returns of natural foraging in a patch of food: the longer the bear spent in the 'patch' (manipulating the feeder), the lower the rate of food intake as food becomes depleted. When the cost of foraging (manipulating the feeder) equals or outweighs the benefits of the food reward, the bear should abandon the feeder (Charnov 1976; Rosenzweig 1974). The density of food remaining when a black bear gave up on the feeder was the GUD. A lower GUD indicated that the forager has spent more time in the patch and thus they perceived a more desirable food reward (Garb 2000). Our feeders were designed to produce a GUD, allowing us to compare the black bears' preferences for an anthropogenic food, deer corn, to their natural forage, acorns or berries.

Our feeders are modified Amazing Graze™ horse toys (Horseman's Pride, Millsburg, OH), a durable, cylindrical plastic toy about 50 cm in length and 15 cm in diameter, with a 7.5-cm hole in the side to allow treats to be dispensed as the toy is rolled. We drilled five holes 3.5 cm in diameter in one end of the feeder, allowing food to be dispensed when the feeder is hung vertically and shaken or batted (Fig 3.1). To standardize the size of the food reward with respect to number of calories and to make it more challenging to get out, we packed food items in edible, biodegradable gelatin pill capsules 3 cm in diameter and 7.5 cm long (Torpac, Fairfield NJ; Fig 3.2).

To compare among food types, it was important that the bears did not completely deplete the food from the feeder before giving up. Therefore, to increase the difficulty of obtaining a food reward, we mixed the food capsules with larger, empty gel capsules (3.5 cm in diameter and 11 cm long) that would not fit through the feeder holes.

Zoo testing

We tested the feeder design with captive grizzly bears (*Ursus arctos*) in collaboration with the Tulsa Zoo in Tulsa, Oklahoma. The feeders were given to 2 female grizzlies weighing about 180 kg, comparable in size to a large male black bear. The food capsules used in the feeders were filled with "kibble" intended for large omnivores, provided by the zoo. We gave the feeder to the grizzly bears without any tethering to test their durability. For subsequent tests, the feeders were suspended from eyebolts in the bear enclosure with 2 cm chain and hung about 1.5 m above the ground, requiring the bears to sit on their hind legs and use their front paws to manipulate the feeders. We exposed the feeders to the bears for 1-hour intervals and recorded the

number and rate at which capsules were obtained, and the bears' interactions with the feeders and capsules.

The most successful design used 30 small capsules containing kibble and 15 large empty capsules acting as a substrate. With this combination, the grizzlies were able to extract and consume food capsules, but abandoned the feeder after 40 minutes with food still remaining (Fig 3.3). Because the captive grizzlies were under no foraging constraints, the fact that they would eventually abandon the feeder suggested that wild black bears, which may need to maximize energy intake rates, would do the same when the rate of gain from the feeders decreased to or below the rate of gain from natural foraging.

The feeders proved to be extremely durable; after multiple trials in the grizzly enclosure, they were scuffed and scratched but not structurally damaged. The grizzly bears tossed them around, shook and rubbed them against enclosure fences and fixtures, and attempted to crush them when they were on the ground, but they did not significantly damage the body.

Field methods

Feeding trials were divided into three seasons: early summer (3–28 June 2015) late summer (29 June–12 August 2015), and autumn (18 September–24 October 2015). These time periods coincided with changes in the natural diet available in the wild: acorns from the previous autumn in early summer, soft mast (primarily blackberries) in late summer, and fresh acorns in autumn. We deployed feeders in the field in pairs. One feeder contained capsules filled with deer corn, the anthropogenic diet item most widely used by bears in the study area. The other feeder contained the staple natural diet item

available at the time: hard mast (acorns) in early summer and autumn, and soft mast (blueberries) in late summer.

Each feeder contained 30 food capsules and 15 larger substrate capsules. We standardized size of the food reward by calorie, so that each food capsule contained about the same number of calories regardless of food type (Table 3.1). To ensure the bears could smell the diet items inside the capsules, each feeder also contained a scent lure consisting of a small amount of that feeder's diet item inside a nylon bag. The scent lure was attached to the inside of the feeder lid and hung near the top of the feeder, so that it could not be removed and would not block any holes or capsules.

We collected fallen red oak (*Quercus rubra*) and white oak *(Quercus alba*) acorns from the study area to match the acorns naturally available during early summer as closely as possible. We used commercial, chemical-free dried blueberries with no sugar added (Bella Viva Orchards, Hughson, CA) for the late summer trials to approximate the berries available at that time period.

We selected feeder sites based on previous knowledge of bear activity from live trapping, hair snare work, and locations of GPS collared bears. Each pair of feeders was suspended from tree branches using 13-mm steel cable, 1–1.5 m off the ground. The two feeders were placed 5–10 m apart. Due to the proximity of the 2 feeders, we assumed that if a bear interacted with one feeder, it was aware of the other feeder at that site.

Feeders were monitored using Stealth Cam model STC-G42NG motion activated cameras (Stealth Cam LLC, Grand Prairie, TX) and checked daily for activity during summer, or weekly during autumn. We removed feeders when rain was likely to prevent

the gelatin capsules from dissolving. If one or more bears interacted with a feeder, we opened the feeders and counted the number of food capsules remaining to obtain a GUD for that food type. The feeders were then refilled and re-hung. All trail camera photographs were downloaded for later analysis.

The research conducted on black bears followed the American Society of Mammalogy guidelines (Sikes et al. 2011) and was approved by the Oklahoma State University Institutional Animal Care and Use Committee under permit number AG-13-6.

Data analysis

Each bear-feeder interaction was characterized in 3 ways based on trail camera photographs and the recorded GUD: successful or unsuccessful, paired or single, and unique or multiple individuals. A successful interaction was one in which the bear was successful in obtaining a food reward from the feeder. A paired interaction was one in which the bear interacted with both feeders, and a unique interaction was one in which only one bear interacted with that feeder between times the feeder and camera were checked (about 24 hours). We also categorized interactions as observation or manipulation. An observation interaction was one in which the bear examined or sniffed the feeder but did not manipulate it. In a manipulation interaction, the bear used its jaws or paws to manipulate the feeder in an active attempt to obtain a food reward.

In cases where multiple bears interacted with the same feeder in one night, we examined trail camera photos and the duration of each interaction. If only one of the bears actually manipulated the feeder, we included it as a unique interaction for

analysis. If there were multiple manipulation interactions in a single night, we did not include the trial in the analysis. We also included unpaired interactions assuming that because the feeders were placed close together a bear interacting with one feeder was aware of both. The feeder that was not interacted with was included in the analysis with a GUD of 100%.

We used histograms and a quantile probability plot lot to visualize the data. This analysis indicated a non-normal distribution with a strong right skew. Because the data were a non-normal distribution, we calculated the median GUD of all unique interactions (including and excluding unsuccessful or observational interactions) and used the Wilcoxon Rank-Sum test to compare median GUD of different diet items by season.

Results

During the early summer feeder season, we had 33 trap nights across 3 feeder sites and compared acorns and corn. The late summer season compared berries and corn with 57 trap nights: 18 at the first set of locations and 39 at the second. The autumn season compared acorns and corn in 35 trap nights.

We recorded 20 bear interactions with the feeders during the early summer season. In 13 early summer interactions, bears successfully obtained food from the feeders. Examination of trail camera photos indicated that on 8 nights, only 1 bear interacted with a pair of feeders. These interactions were classified as unique and included in the analysis. Based on trail camera images and natural markings, pre-existing collars, and ear tags, we estimate that 5–7 individual bears were observed in this sample. The median GUD for corn was significantly lower than for acorns (Z =

1.79, $P = 0.04$; Fig. 3.4).

In the late summer season, we recorded 26 interactions, 21 of which were successful. Thirteen of these interactions were classified as unique and included in the analysis. We estimated that 7–9 different individual bears interacted with the feeders. The median GUD of corn was significantly lower than the median GUD of blueberries ($Z = 1.97$, $P = 0.04$; Fig 3.4).

In the autumn season, we recorded 17 interactions, only 6 of which were successful in obtaining food from feeders. We recorded 10 unique manipulation interactions, all of which were included in the final analysis. Autumn had the fewest unique bears; we estimated that 3–5 individuals were represented in 17 interactions. Median GUD of corn and acorns was not significantly different ($Z = 0.71$ $P = 0.48$; Fig 3.4) in autumn, due in part to the very low number of successful trials.

In early summer, we recorded three non-unique interactions bringing the total interactions to 11, and in late summer, we recorded two non- unique interactions, bringing the total to 15 interactions. Including non-unique interactions, in which more than one bear interacted with the feeder in the same night, did not change the statistical outcome of the tests (early summer $Z = 2.527$, $P = 0.012$; late summer $Z = 2.583$, $P = 0.010$).

The duration of each interaction provides an alternative measure of effort expended on the feeders (Table 3.2). Two interactions included in the GUD analysis were removed from duration analysis due to inaccurate or unreadable time stamps. One obvious outlier, a 72-minute interaction with an acorn feeder in which only one capsule was obtained, was removed from the autumn duration analysis. This bear was very

small, and likely an inexperienced forager who had difficulty reaching the feeder to manipulate it.

Although bears invested more time per interaction in manipulating feeders with corn than feeders with blueberries or acorns in all 3 seasons, none of the differences were statistically significant (early summer $t = 0.78$, $P = 0.23$; late summer $t = 0.68$, $P = 0.25$; autumn $t = 0.83$, $P = 0.21$). The mean duration of autumn interactions with both food types was significantly lower than the mean duration in early summer (natural foods: $z = 3.11$, $P = 0.002$; corn: $z = 2.70$, $P = 0.003$) or late summer (natural foods: $z = 5.25$, $P < 0.0001$; corn: $z = 6.88$, $P < 0.0001$). Amount of time invested during a feeder interaction varied widely among individual bears. Duration of interaction was loosely correlated with the number of capsules obtained during that interaction ($F = 3.43$, $R^2 = 0.12$, $P = 0.02$; Fig. 3.5), but amount of time and number of capsules obtained during a single interaction varied widely.

Discussion

The lower GUDs for corn versus natural foods, when equally abundant and requiring equal effort, suggests that this anthropogenic food was a preferred food resource for black bears. The bears may be expressing a preference for corn, a high fat and high protein food source that is widely available in their recolonized habitat.

Alternatively, the black bears may have interacted more with the feeders containing corn because they are used to manipulating deer feeders, somewhat similar to our experimental feeders, to obtain corn. The natural diet items offered in the feeders were abundantly available in the natural habitat, and require a very different foraging strategy to obtain. Black bears may have interacted with natural food feeders less and

given up earlier because they were familiar with obtaining those items in natural surroundings, rather than from an artificial feeder. In either case, our data indicate that bears expended more time and effort to obtain corn rather than obtain natural foods from our experimental feeders, and diverted time away from foraging for natural food resources. In autumn 2015, when acorns became superabundant (D. Techentin, pers. commun.) in eastern Oklahoma, bears spent very little time manipulating the feeders, suggesting that effort involved in extracting food from the feeders was less profitable than foraging on naturally abundant foods during the period of hyperphagia. At this time of year, bears (especially females) are focused on gaining weight before denning for the winter, and efficient foraging becomes a necessity. Despite the higher fat and calorie content of the corn in the feeders, during this period the bears largely ignored them to focus on eating the abundant acorns, which could be obtained by grazing rather than time consuming manipulation.

In the 10–15 years since they began recolonizing the region, the Oklahoma Ozark bears have adapted their food habits to include anthropogenic foods. This is consistent with other studies showing that black bears are highly adaptable to changes in food availability (Fortin et al. 2013) especially in human-dominated areas where anthropogenic food sources are abundant (Ditmer et al. 2016; Greenleaf et al. 2009). Bears in urban or populated environments will quickly recognize novel sources of food such as feed and garbage, and incorporate them into their foraging patterns (e.g., Baker et al. 2008; Beckmann and Berger 2003; Merkle et al. 2013). Bears in human-dominated areas also tend to shift their patterns of activity, becoming more active in evening and after dark (Beckmann and Berger 2003).

Our results indicate that presence and availability of natural forage are not complete deterrents to bears using corn as a food resource, although it may lessen the likelihood of damage. In autumn when acorns were extremely abundant, we recorded fewer and shorter interactions with corn and acorn experimental feeders. We recorded the most interactions with feeders in late summer, a common period of low food availability in black bear habitat when summer berries have withered but the hard mast crop has not yet ripened (Inman and Pelton 2002). Rates of interaction were lowest during periods of high caloric density in autumn when hard mast was abundant. This is again consistent with other studies indicating that bears may abandon supplemental feed when natural forage is available. In Washington, black bears used supplemental feeders provided to mitigate tree damage in early spring following their den emergence and abandoned the feeders when berries ripened (Ziegltrum 2004). Our results suggest that it is not likely that bears will entirely abandon anthropogenic foods like corn. The Ozark region black bears are well adapted to an urbanized environment where deer feeders are ubiquitous, and our data indicates that they have incorporated corn into their diet as a staple source of food.

Potential nutritional consequences

Black bears incorporating corn as a staple part of their diet will likely not experience serious negative nutritional consequences. The corn used in deer feeders is high in fat and protein and easily digestible, and it is not likely to have serious impact on bears incorporating it as a large part of their diet. The high digestibility of corn and the laxative effects of fiber may lead to an increased passage rate and less efficient digestion in bears consuming large amounts of corn, but this is not likely to be

detrimental unless the bears include very little else besides corn (Gray 2003). However, corn is not a good substitute for important nutritional components of acorns, especially for reproductive females. There is a high correlation between hard mast availability and rates of reproduction, cub survival, and median age of primiparity, making acorns an important resource for bears preparing for denning (Inman and Pelton 2002). This may explain why the Ozark region bears abandoned experimental feeders to focus on consuming acorns from the large 2015 mast crop in eastern Oklahoma. Spring acorn abundance has a significant impact on milk fat content in lactating mothers (McDonald and Fuller 2005), but our bears still demonstrated preference for corn, which is lower in fiber but higher in fat, in early spring (Gray 2003). This may be due to the overall poorer quality of the acorns available in spring, because they are left over from the previous autumn's mast crop. The autumn acorns were fresh and of higher nutritional quality than spring acorns. The acorns available in spring are much older, and their quality has been degraded by exposure and insect damage. The majority of acorns remaining in the spring are also red oak acorns, which are less palatable due to their high tannin content. The importance of nutritional quality of natural forage highlights the need for a study of forage availability in this area to evaluate diet options available without supplemental feed from deer feeders and other anthropogenic sources

The amounts of corn obtained from feeders varied widely between individual bears; nutritional consequences will vary as well. The high nutrition content of corn means that bears incorporating it into their diets is not likely to be harmful (Gray 2003), and in many cases may be beneficial. In Minnesota, use of crops as a staple food source was positively correlated with weight and fat in both sexes, as well as the physical

stature of male bears (Ditmer et. al 2016). Male bears' preference for high-calorie food make them willing to range further and be less wary of threats, which can lead to significant conflict and damage (Ditmer et al. 2015.) The potential for increased human-bear conflict caused by bears damaging deer feeders is likely to reduce human tolerance for coexistence with bears (Bowman et al. 2001), a serious risk for the population re-establishing in this human dominated landscape. Further research into the forage availability and diets of the bears in this region is needed to understand the nutritional consequences of bears incorporating deer corn into their diet.

Benefits of giving up density methodology

Giving up density is a useful methodology for evaluating dietary preference, but it has never been adapted for use in large, omnivorous species. The majority of dietary research in black bears has been scat surveys (e.g. Dobey et al. 2005; Fortin et al. 2013) and stable isotope analyses (e.g. Hopkins et al. 2012, Rode 2016), or land use surveys (e.g. Ditmer 2016) that provide little information as to dietary preference versus dietary use. As generalist omnivores, black bears are well suited to recolonizing and adapting to urban or human-dominated environments. As they expand back into parts of their historical distribution that are now human dominated, property damage by foraging bears becomes a major source of conflict. In the Ozark region, damage to deer feeders is already a significant source of conflict, and a better understanding of the dietary preferences motivating bears to forage for anthropogenic foods is necessary to mitigate potential conflict. Based on our results, for example, curtailing deer feeding until autumn, when acorns fall, might reduce considerable damage to deer feeders.

In this study, we combined GUD methods with trail cameras, which provided

more accurate data than cafeteria style studies. Monitoring feeders with trail cameras gave us the duration of each interaction, and allowed us to be certain that each giving up density data point is associated with only one bear. The large number of individually marked bears, tagged as part of a larger study, assisted with identifying individuals.

Populations of generalist omnivores are rebounding as they adapt their life styles and habits to human-dominated landscapes. New adaptations to a highly altered landscape require new tools to study how large animals with omnivorous diets incorporate anthropogenic food sources into their diets. This experimental design provides an understanding of the behavioral choices that motivate diet preference and foraging strategies of recolonizing bears. This information will be valuable to wildlife managers making decisions and recommendations to mitigate human-bear conflict.

Figure 3.1 A young female bear interacting with a hanging feeder. Holes drilled in the bottom allow gelatin capsules containing either natural food or deer corn to be shaken

from the feeder.

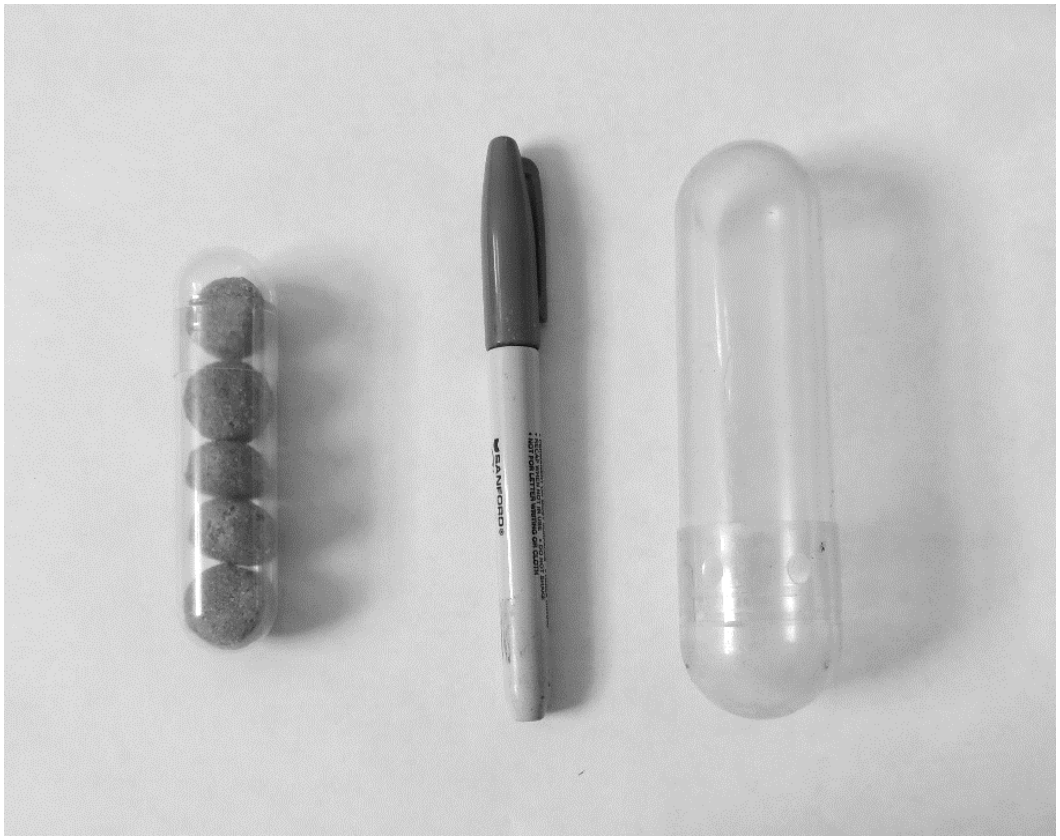

Figure 3.2 Gelatin capsules used to partition food reward (left) and large substrate capsules (right) with a marking pen for scale.

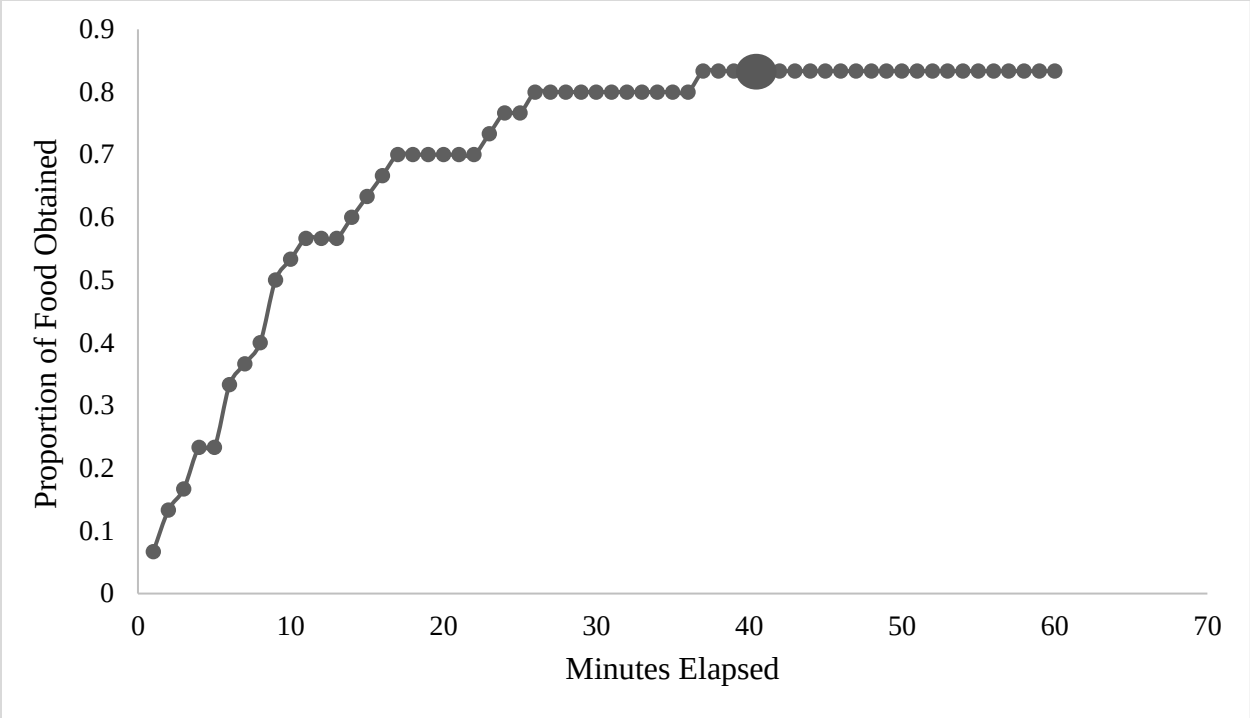

Figure 3.3 Proportion of food obtained from a hanging feeder by a captive grizzly bear. Data points represent the proportion of total capsules obtained in each minute. Large circle represents the time at which the bear abandoned the feeder, after 40 minutes with 83% of capsules obtained (giving up density of 0.17)

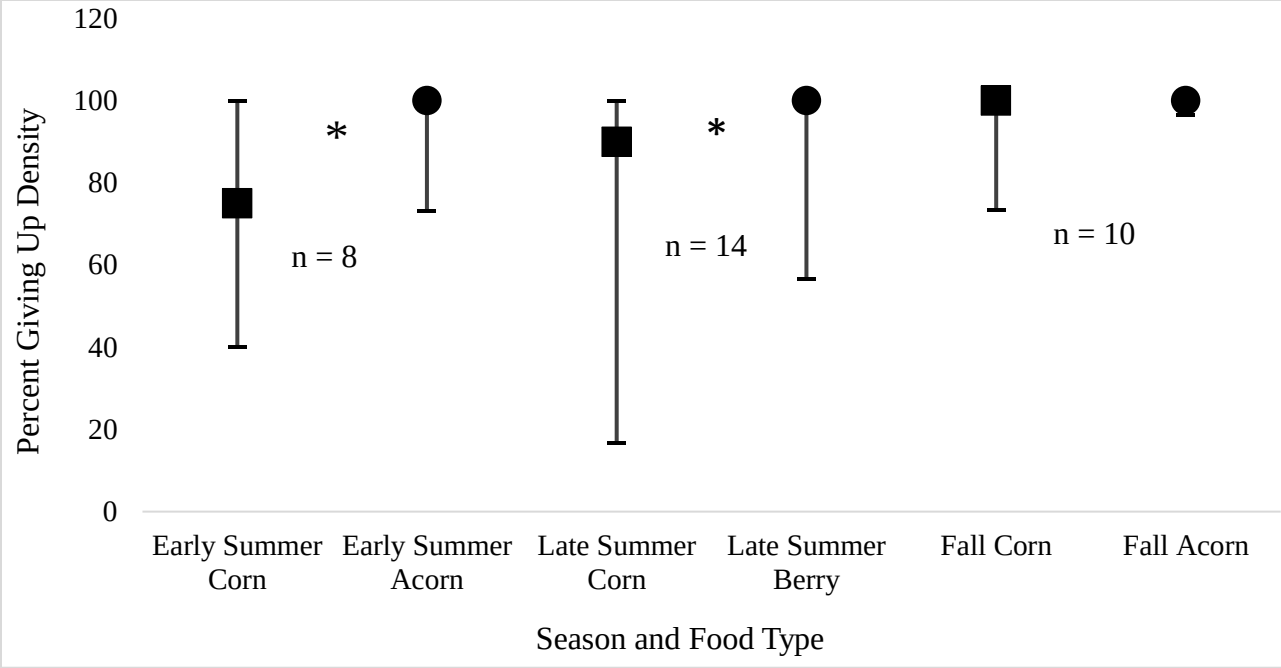

Figure 3.4 Median and range of capsules remaining for each food type across seasons. ■ represents median capsules remaining for anthropogenic foods (corn); ● represents median capsules remaining for natural foods (acorns or berries)

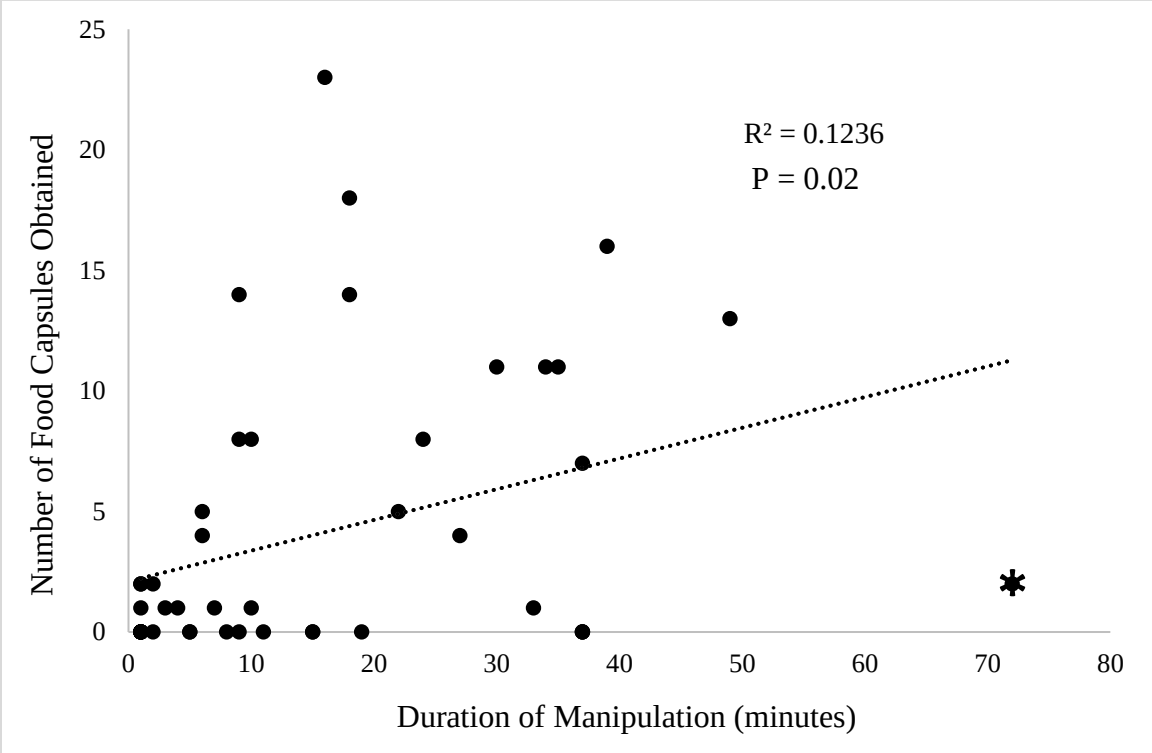

Figure 3.5 Number of capsules obtained during each unique feeder manipulation versus the duration in minutes of that manipulation. All three seasons of feeder trials conducted in 2015 are included. * represents an outlier data point which was removed from the final analysis.

Table 3.1 Contents of food capsules in giving up density trials of food preference in wild black bears in eastern Oklahoma, summer-fall 2015.

Diet Item	kCal/g	g/capsule	kCal/capsule
Acorns	3.87	18	61.2 (60–65)
Corn (early summer and fall)	3.4	15.5	59.98 (60–65)
Berries	3.5	8	28 (25–30)
Corn (late summer)	3.4	8	30 (25–30)

Table 3.2 Mean duration (in minutes) of bear interactions with feeders of each food type by season

Season	Sample Size	Duration	Standard Error
Early Summer			
Natural (acorns)	8	15.62	15.03
Anthropogenic (corn)	8	20.37	15.09
Late Summer			
Natural (blueberries)	13	10.61	18.14
Anthropogenic (corn)	13	14.15	13.92
Fall			
Natural (acorns)	9	3.5*	3.5
Anthropogenic (corn)	9	5.11	6.39

CHAPTER IV

CONCLUSIONS

This black bear population is likely still colonizing the Ozark region of eastern Oklahoma. The low abundance (an estimated 100.91 individuals ± 23.45) and a preponderance of young males are characteristic of the demography found at the peripheral edge of a range expansion; male black bears are long-distance dispersers, while recently independent females tend to establish a home range close to their mother's home range (Swenson et al. 1998). Many of the young males that make up most of the Ozark region population are likely recent immigrants from the larger Arkansas population. The sex ratio will likely approach a 1:1 ratio, as females gradually immigrate into the region and female cubs born in the area reach adulthood and establish their own home ranges (Bales et al. 2005; Pfander 2016; Swenson et al. 1998).

The low number of adult females currently in this population means that a hunting season could pose significant risk to the population's growth. Of the 49 black bears live trapped in the Ozarks since 2011, only 13 were female, and only 5 of those females were reproductive adults aged 4 years or older. While the abundant young males could likely be harvested without effect, the harvest of any adult female would have a significant, and negative, impact on the population's overall growth and

fecundity.

Management decisions made using these data should note that this abundance estimate is likely biased high. Overestimation due to allelic dropout is a common problem in genetic mark-recapture studies (Creel et al. 2003; Mills et al. 2000; McKelvey and Schwartz 2004a; Waits et al. 2001; Waits and Paetkau 2005) because the failure of amplification at one locus can lead to samples with matching genotypes appearing unique.

There is also a potential for bias due to our relatively low rate of recapture, which was likely a consequence of the small geographical area and comparatively low number of samples. While our 56% amplification success rate is on par with similar black bear genetics analysis (Brown 2008, Gardner-Santana 2007), the small scale of this study meant that we were limited to only 85 samples scored at 7 or more loci.

Finally, we initially conducted our abundance estimate assuming a closed population, based on the small geographic area and relatively short time frame of the study (Brown 2008). However, the rates of immigration indicated by the demography of the population means that our estimate may violate the assumption of no immigration required by closed capture estimates. Due to the short span of the study and the short amount of time between captures and recaptures, immigration is likely to be relatively low. However, immigration is occurring, and violation of the closed population assumption can also lead to an over-estimation of abundance.

The sparse population outside of the core area means that additional hair snare studies with an expanded sampling grid would likely not yield enough samples to be

worthwhile. Analysis of study design for non-invasive sampling has shown that, especially in low-density populations, longer sampling periods are more effective at collecting higher number of samples than more snares (Wilton et al. 2014). Adaptive placement of snares, rather than strict systematic placement, is also a better approach for sparse populations or elusive species (Belant et al. 2011). The high number of marked bears in the population also make camera trap studies with an extended range a viable option for further surveys.

The Ozark bear population is recolonizing a human-dominated landscape and demonstrates the ability to quickly adapt its behavior to a fragmented landscape with abundant anthropogenic sources of food. Black bears' ability to expand their foraging habits and dietary preferences to include anthropogenic sources of food is well documented (Ditmer et al. 2015; Ditmer et al. 2016; Beckman and Berger 2003; Baruch-mordo et al. 2008; Baruch-mordo et al. 2014), and the young population establishing in the Ozarks makes little distinction between natural and anthropogenic foods. In early and late summer, the Ozark bears demonstrated a significant preference for high-calorie, high-fat corn over native acorns and berries. However, in autumn, when fresh acorns became abundant, the bears demonstrated little interest in our experimental feeders, preferring to forage on the locally abundant and easily obtained acorns.

Our findings follow a common trend of black bear populations in human dominated areas: bears will take advantage of the anthropogenic sources of food that are available year-round when their natural foods are scarce but return to natural foraging when those resources are widely available (Baruch-mordo et al. 2008; Baruch-Mordo et al. 2014; Ditmer et. al 2016; Ziegletrum 1997). However, the fact that the Ozark region

bears demonstrated a preference for corn even when a natural food type was equally available indicates that many bears may continue to exploit deer feeders as a source of food, regardless of the amount of natural forage available. Young male bears, who make up the majority of this population, are most likely to exploit high-calorie anthropogenic food sources, and are less wary of potential danger and novel foods (Ditmer et al. 2015; Ditmer et al. 2016). The large number of young male bears in this population have the potential to cause a great deal of property damage, raising tensions and rates of human-bear conflict in the region.

Management Recommendations

Due to the limited spatial distribution and the demographical bias toward young males, a potential hunting season would need to be limited and tightly controlled. There are very few reproductively aged females in the Ozark region, so the harvest of an adult female could be very harmful to the reproductive potential of this population. A limited hunting season could occur in the small area of high bear density on the Cookson WMA. Nevertheless, because even a small harvest could have a serious negative impact on this small and demographically skewed population, it would be extremely important to consider population demographics alongside population size in any management decisions.

Further monitoring of this population and its demographics, through non-invasive means as well as traditional live capture and satellite collar studies, is vital for good management, especially if a hunting season is opened in the region. A detailed habitat analysis is also needed to determine the resources and area available for this growing population, and a potential ecological carrying capacity. However, the

population is likely to reach a social carrying capacity before it reaches an ecological one: growing rates of human-bear conflict will require mitigation. As the population continues to grow and integrate with the human population in the area, outreach and education will become necessary to raise public awareness of the black bear population, and how to safely cohabitate with them.

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VITA

Emily Artz

Candidate for the Degree of

Master of Science

Thesis: GENETIC MARK-RECAPTURE POPULATION ESTIMATE AND DIETARY PREFERENCES IN A RECENTLY RE-ESTABLISHED AMERICAN BLACK BEAR POPULATION

Major Field: Wildlife Ecology and Management

Biographical:

Education:

Completed the requirements for the Master of Science in wildlife ecology and management at Oklahoma State University, Stillwater, Oklahoma in July, 2016.

Completed the requirements for the Bachelor of Science in wildlife ecology at Iowa State University, Ames, Iowa in 2012.

Experience: Research assistant and technician in entomology and pollinator ecology at Iowa State University (2012-2013); US Fish and Wildlife Land Management Research and Development intern (2011); Undergraduate research in seed dispersal by American bison and their role as keystone species in a tallgrass prairie.

Professional Memberships: The Wildlife Society, Central Plains Society of Mammalogy, American Society of Mammalogists, Sigma Xi, Oklahoma State University Natural Resource Ecology and Management Graduate Student Organization; Oklahoma State University Graduate and Professional Student Government Association.